

1 Sensitivity of peatland litter decomposition to changes in 2 temperature and rainfall

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24 Abstract

25 Changes to climate are projected over the next 50 years for many peatland areas. As
26 decomposition of peat-forming vegetation is likely to be intrinsically linked to these changes in
27 climate, a clear understanding of climate-peat dynamics is required. There is concern that
28 increased temperature and decreased precipitation could increase the rate of decomposition
29 and put the carbon sink status of many peatlands at risk, yet few studies have examined the
30 impact of both climatic factors together. To better understand the sensitivity of peatland
31 decomposition to changes in both temperature and precipitation and their interaction, we
32 conducted a short-term laboratory experiment in which plant litters and peat soil were
33 incubated, in isolation, in a factorial design. Treatments simulated baseline and projected
34 climate averages derived from the latest UK climate change projections (UKCP09) for Exmoor,
35 a climatically marginal peatland in SW England. Regular carbon dioxide flux measurements
36 were made throughout the simulation, as well as total mass loss and total dissolved organic
37 carbon (DOC) leached. The largest effect on carbon loss in this multifactor experiment was
38 from substrate, with *Sphagnum*/peat releasing significantly less C in total during the
39 experiment than dwarf shrubs/graminoids. Climate effects were substrate specific, with the
40 drier rainfall treatment increasing the DOC leaching from *Calluna*, but decreasing it from
41 *Sphagnum*. Partitioning between CO₂ and DOC was also affected by climate, but only for the
42 peat and *Sphagnum* samples, where the future climate scenarios (warmer and drier) resulted
43 in a greater proportion of C lost in gaseous form. These results suggest that indirect effects of
44 climate through changes in species composition in peatlands could ultimately turn out to be
45 more important for litter decomposition than direct effects of climate change from increased
46 temperatures and decreased rainfall.

47 **Keywords:** decomposition; Exmoor; blanket peat; uplands; DOC; UK

48

49 1. Introduction

50 Northern peatlands are an important carbon store, holding around one third of the
51 global soil carbon stock (Gorham, 1991). For peatlands to accumulate organic matter, and thus
52 sequester carbon from the atmosphere, the overall loss of carbon from the system through
53 the combined decomposition of plant litter and peat must be lower than C input via litter
54 production and vascular plant root exudation (Frolking et al., 2010; Limpens et al., 2008).
55 Decomposition is the breakdown of organic matter. It can be due to one or a number of
56 physical, chemical or biological processes. Carbon dioxide (CO₂), dissolved organic carbon
57 (DOC), nutrients and stable hummus are among the principal final products of decomposition
58 (Bragazza et al., 2009). Carbon is primarily lost as CO₂ in peatlands (Billett et al., 2010).
59 However, additional losses in the form of methane (CH₄), produced during anaerobic
60 decomposition, and DOC to aquatic systems can be an important component of the peatland
61 carbon balance in some areas (Köehler et al., 2011). The speed and final product of the
62 decomposition process is dependent on the chemistry of the organic matter, and the
63 environment in which it decomposes (Limpens et al., 2008).

64 Understanding how peat and plant litter decomposition could be affected by climate
65 change, specifically changes in temperature and rainfall, is key to an improved understanding
66 of how the peatland carbon source/sink status could change during the 21st century. Northern
67 peatlands are typically dominated by remains of mosses of the genus *Sphagnum*, a bryophyte
68 (Turetsky, 2003). *Sphagnum* has played an important role in peatlands becoming such a large
69 carbon sink. Not only does the recalcitrance of its litter mean that it keeps C in the system for
70 longer, but the presence of *Sphagnum* maintains acidic, nutrient poor conditions which make
71 it difficult for plants with more labile litter to establish themselves (van Breemen, 1995).

72 Climate change may alter vegetation types in peatlands with vascular plants becoming more
73 dominant (Buttler et al., 2015; Dieleman et al., 2015; Fenner et al., 2007; Weltzin et al., 2003).
74 However, many ongoing catchment management programmes aim to encourage *Sphagnum*
75 species over vascular plants as part of peatland restoration programmes (Grand-Clement et
76 al., 2013).

77 While some authors have studied the effects of climate variables on the treatability of
78 water from different peatland plant species (e.g. Ritson et al., 2014; Tang et al., 2013), little
79 work has so far been conducted on the total flux of carbon and the partitioning between
80 gaseous and fluvial losses from different peatland plant species. This partitioning is an
81 important parameter to include in carbon cycle models, particularly for climate change
82 modelling as there is likely to be a lag between DOC release from the soil and its incorporation
83 into the atmospheric pool, as in-stream processing can lead to the temporary storage of C
84 within the aquatic system (Hope et al., 1994). An understanding of partitioning of carbon
85 losses between aquatic and gaseous fractions from different vegetation sources is also
86 important for predicting whether catchment management programmes aimed at restoring
87 certain species have the desired holistic effects of improving drinking water quality (including
88 enhanced C sequestration) and do not result in environmental problem-shifting.

89 Decomposition of litters in the field is traditionally measured using the litter bag
90 technique (Bragazza et al., 2009; Wieder and Lang, 1982). Litter of a known mass is enclosed
91 in mesh bags with openings large enough for decomposers to access the food source, but
92 small enough to prevent the physical loss of litters. Samples are incubated in the field, either
93 on, or just below the soil surface (Johnson and Damman, 1991; Moore et al., 2007) and
94 decomposition rate is quantified via mass loss or changing nutrient quotients of the litter
95 (Keuskamp et al., 2013). *In situ* litter decomposition studies, using this technique, have

suggested multiple drivers for increased decomposition rates in peatland systems including: elevated nutrient additions (Bubier et al., 2007), water-table drawdown (Straková et al., 2012), litter quality (Limpens and Berendse, 2003) and temperature (Moore et al., 2007). However, these field-based studies are limited to a single measure of decomposition; mass loss. Mass loss gives a valuable measure of net decomposition, which can be useful for comparing between litters, but does not provide information on the final product of the decomposition process. Carbon dioxide and DOC not only represent different flux pathways (gaseous versus aquatic), but can also be indicative of different processes and stages of decomposition. CO₂ is in effect representative of fully utilised carbon, whereas DOC could be stored in the peat column or utilised by microbes and exported to the atmosphere via respiration (Pastor et al., 2003; Turetsky, 2003).

Laboratory incubations of peatland litters to examine the relationship between decomposition and climate have been limited to date. Studies that have been conducted have tended to concentrate on a single measure of decomposition, have focussed solely on peat rather than vegetation, or have just looked at one environmental variable, such as temperature (e.g. Neff and Hooper 2002; Moore et al. 2008) or water table (e.g. Freeman et al. 1993), in isolation. The effect of interactions between changes in temperature and rainfall on the decomposition of peatland litter is not currently well understood (Breeuwer et al., 2008). A gradient based mesocosm study measured increased C soil respiration in mesocosms transplanted to a warmer, drier location (Bragazza et al., 2016). However, isolating the effects of climatic variables such as temperature and rainfall from other confounding variables can be difficult in the field as these factors naturally co-vary. It is also difficult to separate the effects on different litters and measure the relative decomposition from different sources in the field or in intact mesocosms. To address this, previous studies have incubated soils or

litters from wetland sites in isolation, notably Moore and Dalva (2001), Neff and Hooper (2002) and Wickland et al. (2007). This was the approach taken here.

A better understanding of the sensitivity of different peatland litters to changes in temperature and rainfall regime is needed to enable predictions of the impact of climate change on the stability of peatland carbon stocks, and to support better parameterisations of models that simulate past, current and future climate (Frolking et al., 2010; Smith et al., 2010). Therefore, the objectives of this research were as follows: (1) assess how litter decomposition rates varied between common ombrotrophic peatland plant species (*Calluna vulgaris*, *Molinia caerulea*, *Sphagnum* moss, and mixed litter) compared to peat; (2) determine the relative importance (or partitioning) of gaseous (i.e. CO₂) versus aquatic (DOC) fluxes during decomposition with respect to plant species and peat; (3) evaluate the importance of simulated temperature and rainfall changes on controlling total decomposition, C fluxes and partitioning between CO₂ and DOC. This final objective links to another study (Ritson et al., 2014), which used the same experimental treatments to examine the impact of climate change on the treatability (for potable water supply) of dissolved organic matter.

135

2. Methods

2.1. Field site and sample collection

In the south-west UK, blanket peat covers large parts of the upland areas and constitute an important store of C (Parry and Charman, 2013). These blanket peatlands represent the southern-most blanket peat in the UK and are considered climatically marginal (Clark et al., 2010). The shallow peat of Exmoor was heavily damaged by intensive drainage during the 19th and 20th centuries, and recent efforts have been made to restore large areas

143 through a programme of ditch blocking (Grand-Clement et al., 2014, 2013). The field sites are
144 dominated by *Molinia caerulea* (purple moor grass) (Gatis et al., 2015) and are classified as UK
145 National Vegetation Classification M25: *Molinia caerulea* - *Polentilla erecta* mires (Rodwell,
146 1991). Paleoecological studies indicate that purple moor grass expansion corresponds with
147 the industrial revolution and prior to this, *Sphagnum* was more dominant species within this
148 region (Chambers et al., 1999). Exmoor typically receives precipitation of around 1800–2600
149 mm yr⁻¹ with mean winter and summer temperatures of between 4.5–5.5 °C and 10–12 °C,
150 respectively (Met Office, 2014). Exmoor receives a relatively high input of nutrients through
151 atmospheric deposition, and the mean deposition of total nitrogen and oxidised sulphur for
152 the period 2008–2012 was estimated to be 15.5–16.16 kg N h⁻¹ yr⁻¹ and 4.41–4.86 kg S h⁻¹ yr⁻¹,
153 respectively (DEFRA, 2015).

154 Samples of vegetation and peat were collected from two catchments, Aclands (51°
155 07'54.2" N 3° 48'43.3" W) and Spooners (51° 07'23.3" N 3° 45'11.8" W), within Exmoor
156 National Park, UK, during July 2013. A further description of these field sites, including maps,
157 can be found in Grand-Clement et al. (2014) and Luscombe et al. (2015). Five different
158 substrates were collected: *Calluna*, *Molina* and *Sphagnum*, mixed litter and peat. Fresh leaves
159 and branches were collected from *Calluna vulgaris* and *Molinia caerulea*, avoiding any plants
160 which had been partially eaten by herbivores. Mixed litter (predominantly *Molinia caerulea*
161 but also some *Eriophorum vaginatum*, both at the early stages of decomposition) was collected
162 from the bog surface. Intact *Sphagnum* spp. sods (entire plants) were collected from the
163 centre of stands and peat was collected from the top ca. 30 cm of the soil profile using a screw
164 auger. The samples were transported back to the laboratory in cool boxes within 8 hours of
165 collection, and stored at 4 °C prior to their preparation. Sample collection is described fully in
166 Ritson et al. (2014).

167

168 2.2. Sample preparation

169 Samples from each vegetation class (*Calluna vulgaris*, *Molinia caerulea*, mixed litter
170 and *Sphagnum* spp.), were cut into 2-3 cm sections and mixed thoroughly within their class.
171 Any material not from the target group was removed. Once sorted, cut and homogenised, the
172 vegetation samples were left to air-dry in the laboratory to constant mass (Moore et al., 2007).
173 The structure of peat is permanently altered when dried, becoming hydrophobic (Holden and
174 Burt, 2002). For this reason, the peat was treated differently to the other substrates and was
175 kept refrigerated and at field moist conditions prior to the experiment.
176 Air-dried sub-samples of each plant species and field moist peat ($n = 5$) were oven dried at 70
177 °C for 24 h to determine an air-dry to oven-dry conversion factor. This conversion factor was
178 applied to estimate the oven-dry weight of samples at the beginning of the simulation. These
179 oven-dry samples were analysed for litter quality (see methods below; Table 1).

180

181 2.3. Experimental design

182 2.3.1. Experimental set up

183 Polypropylene Buchner funnels (Thermo Scientific, USA) were filled with
184 approximately 2 g dry-weight of air-dry vegetation or 10 g homogenised peat. The stem of the
185 funnel was packed with acid washed glass wool to ensure no loss of particulates, and each
186 Buchner funnel was placed above an amber-glass bottle to collect leachate, for separate
187 analysis (see Ritson et al., 2014). The climate control facilities comprise two versatile
188 environmental test chambers (Panasonic MLR-352H). The incubators were set to cycle
189 between 12 h at the mean daily maximum and 12 h at the mean daily minimum temperature.

Deionised water was applied in amounts equivalent to the monthly rainfall, with a dispenser set to the required volume based on the area of the Buchner funnel top. This was applied gradually and evenly, over approximately 30 seconds, taking care not to disturb the samples.

2.3.2. *Experimental design*

The experiment was set up as a factorial ANOVA design, where substrate, temperature and rainfall were the treatment factors in a 5 x 2 x 2 design. Five replicates of each treatment were conducted, giving a total of 100 samples (see Ritson et al., 2014). Climate treatments were based on (i) baseline (1961-1990; Met Office, 2013) mean minimum and mean maximum daily temperature and mean rainfall totals for July and August for Exmoor National Park, UK, and (ii) a projection from the UKCP09 high emissions scenario for 2080s (2070-2100). All four possible combinations of baseline and projected temperature and rainfall were applied as treatments. Mean minimum daily temperatures were set to 11.0 °C (± 0.3 °C for test chamber error) for the baseline, and 15.4 °C (± 0.3 °C) for the UKCP09 treatment. Mean maximum daily temperatures were set to 18.2 °C (± 0.3 °C) for the baseline treatment and 23.7 °C (± 0.3 °C) for the UKCP09 treatment. Rainfall totalled 100.9 and 117.5 mm for the baseline treatment for July and August, respectively, and 75.4 and 67.2 mm for the UKCP09 treatment. Overall, this was equivalent to a ca. 5.0 °C increase in temperature and ca. 35 % decrease in rainfall for the UKCP09 treatments. The relative humidity (to program the test chambers) and number of rain days were set at mean values from the baseline period for all samples (for relative humidity: 80 % in July, 79 % in August; for rain days: 12 in July and 14 in August). The rainfall total for each month was split evenly between the rain days (resulting in irrigation amounts ranging between 4.2 mm and 8.4 mm per event, or 19 and 38 ml, depending on treatment and month),

213 and the amounts were applied every 2-4 days. The conversion from mm to ml was based on
214 the area of the funnel.

215

216 2.4. Measurements

217 2.4.1. Carbon dioxide evolution

218 At eight points during the experiment (on days 2, 7, 10, 17, 31, 45, 50, and 60 after the
219 start of the incubation) samples were analysed for CO₂ efflux. Samples were taken out of the
220 incubators individually and the top part of the Buchner funnel, containing the sample, was
221 removed and placed into an airtight container (Lock&Lock, South Korea) which was connected
222 to an infra-red gas analyser, IRGA (8100A, LI-COR, USA). Samples were removed for
223 measurement in a random order to avoid the introduction of systematic measurement error.
224 Observations with the IRGA lasted for 2 minutes during which time the concentration of CO₂
225 in the chamber (water vapour compensated) was measured every second, and the samples
226 were out of the incubators for a maximum of 5 minutes. The rate of change in CO₂
227 concentration in the headspace during the 2-minute measurement period was used to
228 calculate the instantaneous flux by applying a linear regression. The IRGA system was allowed
229 to purge for a minute between samples, and a 30 second deadband (a period where the IRGA
230 is running without the measurements being used in the flux estimate) was introduced at the
231 beginning of each measurement to allow for adequate mixing of air in the headspace. The CO₂
232 time series for each sample was analysed separately, and the deadband was adjusted where
233 necessary. All flux estimates were derived from a minimum of 60 consecutive CO₂
234 concentration measurements (i.e. 60 seconds). Individual flux estimates were converted to
235 mg CO₂-C g C⁻¹ day⁻¹. Cumulative CO₂ flux over the course of the two month experiment was

236 calculated for each sample based on a linear interpolation of fluxes between the successive
237 measurements (Neff and Hooper, 2002), expressed as $\text{mg CO}_2\text{-C g C}^{-1}$.

238

239 2.4.2. *Dissolved organic carbon*

240 DOC analysis was performed on the cumulative leachate collected at the end of the
241 simulation period. Samples were filtered through a GF/F filter (Whatman, UK) with a nominal
242 pore size of $0.7\ \mu\text{m}$, acidified to $\sim\text{pH } 2$ with HCl and stored at 4°C prior to analysis. Organic
243 carbon content was measured as non-purgeable organic carbon (NPOC) using a total organic
244 carbon analyser (Shimadzu TOC-V, Japan). Calibration and quality control checks were run
245 daily from freshly prepared standards and samples were analysed within one week of the end
246 of the simulation. Organic carbon concentration (mg L^{-1}) was converted to DOC leached (mg)
247 based on the volume of leachate collected. As temperature and rainfall treatment affected
248 the volume of sample collected, this was then reported as mg DOC-C g C^{-1} , based on the
249 starting dry mass and initial C content of each sample.

250

251 2.4.3. *Mass loss*

252 At the end of the two-month incubation period, samples were removed from the
253 incubators, dried at 70°C for 48 h and weighed. C loss was estimated based on the starting
254 mass and starting C content, as well as the end mass and end C content. This was expressed
255 as mgC g C^{-1} .

256

257 2.5. *Litter quality analysis*

258 The oven-dry subsamples from the beginning, and all samples from the end of the
259 experiment were ground using a laboratory disc mill with an agate barrel (Tema Machinery
260 Ltd, UK) and analysed for total C and N (Thermo FLASH 2000 elemental analyser). The handling
261 and analysis of samples were undertaken together with reference material (Pahokee Peat,
262 International Humic Substances Society) in order to verify the calibration and quality of the
263 results. Subsamples from the start of the experiment were also analysed for C character using
264 nuclear magnetic resonance (NMR) analysis. Cross polarisation-magic angle spinning (CP-
265 MAS) ^{13}C NMR analysis was undertaken at the University of Reading chemical analysis facility
266 (CAF) on a Bruker AV500 instrument based on the method of Wong et al. (2002). This
267 technique is used widely in environmental studies and is good for looking at relative changes
268 in carbon distribution across a series of samples (Simpson et al., 2011). For the vegetation
269 samples 1024 transients (executions of the pulse sequence) were taken, however due to the
270 presence of paramagnetic material in the peat soil transients were increased to 4096 to
271 improve the signal to noise ratio. Magic angle spinning (MAS) spin rate was set at 10 kHz to
272 shift spinning side bands out of the region of interest and all signals were externally referenced
273 to adamant. The aromatic peak between 110-140 ppm was integrated and its percentage of
274 the total signal calculated to give an indication of the differences between relative aromaticity
275 between samples (Peuravuori et al., 2003), which can be used as an alternative qualitative
276 measure of the recalcitrance of the litter (see Table 1). For further reading on using NMR
277 analysis for environmental research, see Simpson et al. (2011).

278

279 2.6. Statistical analysis

280 All statistical analyses were carried out using R version 3.0.2 (R Core Team, 2015) and
281 plots were generated using the ggplot2 package (Wickham, 2009). The experiment involved a
282 three-way analysis of variance (ANOVA) design where temperature, rainfall and substrate
283 were the experimental factors. The effect of these factors, and their interaction, on response
284 variables of CO₂ flux, mass loss, DOC leached and CO₂:DOC ratio production was analysed
285 (Table 2). A Fligner-Killeen test was performed to determine that there was equal variance
286 between groups, and residuals were checked for normality. Carbon dioxide flux, DOC flux and
287 CO₂:DOC ratio were all square-root transformed prior to inclusion, in order to meet the
288 assumption of normally distributed residuals. A critical value of $P = 0.05$ was used for statistical
289 significance. For multiple comparisons, the Tukey honest significant differences (HSD) test was
290 used. The variance explained by each factor was estimated using omega-squared (ω^2), a test
291 which is suitable for estimating effect size from small sample sizes (Keselman, 1975).

292

293 3. Results

294 3.1. Variability of decomposition rates between different substrates

295 Decomposition was greatest for the *Molinia* samples and slowest for the peat samples
296 (Fig. 1). The same substrate rankings were evident in terms of the following: the estimated
297 cumulative CO₂ flux (means ranged from 942.87 mgC gC⁻¹ for *Molinia* to 41.26 mgC gC⁻¹ for
298 peat; Fig. 1); the DOC flux (means ranged from 20.95 mgC gC⁻¹ for *Molinia* to 1.07 mgC gC⁻¹
299 for peat; Fig. 1); and the overall total measured mass loss (means ranged from 511.21 mgC
300 gC⁻¹ for *Molinia* to 27.43 mgC gC⁻¹ for peat; Fig. 2). Substrate was a significant explanatory
301 factor in the ANOVA for all of these response variables ($P < 0.001$; Table 2). Post-hoc tests

suggested significant differences in the order *Molinia* > *Sphagnum* = Mixed litter > *Calluna* > peat for CO₂ flux; *Molinia* > *Calluna* = Mixed litter > *Sphagnum* > peat for DOC flux; and *Molinia* > *Calluna* > *Sphagnum* = Mixed litter > peat for mass loss.

There was a steady decrease in CO₂ release over the entire incubation period, for all substrates and treatments. The decrease in CO₂ flux was greatest for the *Molinia* samples, with a reduction in estimated flux between the first (day 1 to 30) and second half (day 31 to 60) of the experiment of 64.27%, compared to peat where the difference was 25.13% – the lowest decline recorded.

In terms of total mass balance, the overall loss of C measured directly through mass loss of the substrate and by the total fluxes (DOC + CO₂) were not equal. Whilst DOC fluxes were less than the total mass loss, the cumulative CO₂ flux estimated by integration across the experimental period (*sensu* Neff and Hooper, 2002) was greater than the total mass loss.

3.2. Carbon dioxide to DOC partitioning

The cumulative flux of CO₂ was greater than DOC for all of the samples (Fig. 1). Mean CO₂:DOC quotients ranged from 31.56 for *Molinia* to 68.67 for *Sphagnum* (all climate treatments pooled), indicating that proportionally more carbon was lost as DOC from *Molinia* than *Sphagnum*. Substrate was again the strongest factor explaining the variation in partitioning between gaseous and aquatic fluxes ($P < 0.001$). There was a significant relationship between total CO₂ flux and DOC leached between groups when all data were pooled, indicating that DOC leaching increases with CO₂ flux. However, this relationship was not observed within substrate groups.

The general trend was for an increase in the CO₂:DOC quotient under UKCP09 future climate scenarios (Fig. 1). There was a statistically significant interaction between rainfall and

substrate ($P < 0.001$), with the proportion of C lost as CO_2 generally increasing under the UKCP09 (drier) rainfall scenario. Post-hoc tests revealed that drier conditions were only significant in changing the partitioning between gas and aquatic carbon fluxes for the peat ($P < 0.001$) and *Sphagnum* ($P < 0.001$) samples, the mean of which increased from 24.91 (± 5.74) to 78.64 (± 13.85) and from 47.34 (± 2.41) to 101.78 (± 16.25) when comparing the baseline temperature and rainfall to the UKCP09 temperature and rainfall treatments. Partitioning between CO_2 and DOC in other substrates, *Molinia*, *Calluna* and mixed litter, was unaffected by temperature and rainfall treatments ($P > 0.05$).

334

3.3. *Relative importance and interaction between temperature and rainfall in controlling decomposition*

Increasing temperature had a significant effect on decreasing DOC flux ($P < 0.001$) and increasing mass loss ($P < 0.001$). In addition, significant interactions between temperature and substrate ($P < 0.001$) and rainfall and substrate ($P = 0.001$) were seen for DOC flux and for mass loss ($P = 0.007$ and $P = 0.035$, respectively), indicating that the effect of simulated climate change depended on substrate type. For DOC, post hoc tests suggested that the interaction was only significant between temperature and *Calluna*, whereas for rainfall, there was a significant interaction between rainfall treatment and *Calluna* and rainfall treatment and *Sphagnum*. This interaction between rainfall and substrate was not unidirectional, however, with the drier rainfall treatment increasing DOC in *Calluna* samples, but decreasing it in *Sphagnum*. With mass loss, only *Sphagnum* was close to being significant on its own. Differences in cumulative CO_2 flux between climate treatments (both temperature and rainfall) were non-significant ($P > 0.05$), though rainfall was close to the critical threshold ($P =$

0.075). No significant interaction between temperature and rainfall for any of the measures of decomposition was observed (CO₂ flux, DOC flux or mass loss).

4. Discussion

4.1. *Carbon fluxes and decomposition rates between different substrates*

The fluxes of CO₂ and DOC, and the mass loss, were lowest from the peat samples compared to litter sources, consistent with previous work (Moore and Dalva, 2001; Tang et al., 2013). Decomposition in peatland ecosystems progresses along a continuum from freshly senesced plant litter to peat. Part of the reason that peat decomposes more slowly than fresh litter is because more labile compounds are preferentially utilised by decomposers (Wieder and Lang, 1982). This results in older peat being composed of progressively more recalcitrant materials (Broder et al., 2012). In the field, these differences are likely to be even greater as decomposition of peat is further inhibited by anoxic and acidic conditions which prevail for longer at greater depths. Here, we used measures of C:N ratio and aromaticity to characterise the quality of the substrates. A high C:N ratio usually suggests recalcitrant litter, as N is limiting for microbes and can also stimulate the activity of certain enzymes (Bragazza et al., 2012). Here peat had the lowest C:N ratio, which could be due to retention of N containing compounds during the decomposition process (Kalbitz and Geyer, 2002). Aromaticity is another potential measure of decay resistance as aromatic polymers are characterised by structural and chemical recalcitrance. In this instance, peat had the highest estimated proportion of aromatic compounds (Table 1), which could in part explain its low decomposition rate relative to the other substrates.

371 In terms of specific fluxes, *Sphagnum* produced the least DOC in line with previous
372 observations of high retention of carbon in this species (Fenner et al., 2004). *Sphagnum* also
373 produced slightly more CO₂ than *Calluna*, which was unexpected given the well documented
374 recalcitrance of *Sphagnum* litter which has been shown to decompose much more slowly than
375 other peat forming vegetation (van Breemen, 1995). This could have been a result of
376 environmental conditions, specifically water content, as the poor water holding capacity of
377 *Calluna* litter meant that lack of water supply could have limited its decomposition during
378 laboratory simulation experiments (Blok et al., 2015; Toberman et al., 2008).

379 The estimates of cumulative CO₂ flux were approximately double the amount
380 estimated from C mass loss, suggesting an overestimation of cumulative CO₂ flux by
381 integrating point measurements recorded here. This could partly be due to the measurement
382 of some autotrophic respiration from peat roots in the earlier stages of the experiment, or an
383 inadequate representation of night-time respiration rates. However, given that all samples
384 were treated in the same way, and were well mixed to begin with, it is reasonable to assume
385 that the observed differences between groups are robust.

386

387 4.2. Relative importance of gaseous and fluvial fluxes

388 The release of DOC during organic matter decomposition to surface waters is an
389 important component of the peatland carbon budget (Billett et al., 2010). Little is known
390 about partitioning between CO₂ and DOC fluxes in peatlands (Bonnett et al., 2006). Herein, it
391 was shown that *Molinia* produced the most DOC as a proportion of its starting mass, and peat
392 produced the least. However, in terms of the ratio of CO₂ to DOC, *Calluna* and mixed litter had
393 the lowest ratios, signalling that a higher proportion of mass is lost as DOC than CO₂ for these
394 substrates, when compared with the others. Again, absolute values for CO₂:DOC may not be

395 accurate due to overestimation of measured CO₂ fluxes, but comparisons between groups
396 should still be robust.

397 Previous studies have reported DOC fluxes equivalent to over 20% of net ecosystem
398 exchange (NEE) (e.g. Dinsmore et al., 2010; Koehler et al., 2011). In some instances, DOC
399 exported to streams has been estimated to exceed NEE (Billett et al., 2004), highlighting the
400 importance of including this flux when estimating source/sink status of a peatland. Strong
401 correlations have been found between gross primary productivity (GPP) and DOC
402 concentrations in peatland catchments (Dinsmore et al., 2013; Harrison et al., 2008),
403 suggesting a large amount of DOC is produced directly by living plants, indirectly by
404 rhizosphere priming or by the decay of fresh plant litter. The difference in the amount of DOC
405 released from the peat and litter samples in this study during decomposition supports the
406 theory that the decay of fresh litter is a substantive DOC source (Evans et al., 2007; Palmer et
407 al., 2001). In the field, a lot will depend on the regulating effect of water table depth on
408 decomposition, flow pathways of runoff (Wallage and Holden, 2011), as well as substrate.

409 We observed a lower DOC flux but no significant change in CO₂ flux under the UKCP09
410 (warmer) temperature scenario. Combined with the lower DOC flux under UKCP09 (drier)
411 rainfall scenario for some of the substrates (*Sphagnum*), this resulted in an increase in the
412 partitioning coefficient between CO₂ and DOC flux for some of the substrates (*Sphagnum* and
413 peat), which is in accordance with the findings of Neff and Hooper (2002) in their arctic soil
414 samples. This implies that a greater proportion of C released from *Sphagnum* and peat would
415 be lost in the form of CO₂ than DOC under warmer and drier conditions, with the effect of
416 rainfall being slightly stronger than temperature. The strength of the rainfall effect suggests
417 that it is due more to the decreased flux of DOC than due to a reduction in total
418 decomposition. The total DOC measured in this study was in effect net DOC production; a

function of release and transport, as well as its mineralisation. The decreased DOC flux could be explained by a suppression of DOC mobility by drought-induced acidification (Clark et al., 2006; Clark et al., 2012). DOC production could be affected by both microbial enzymatic activity and physical leaching. It could be that DOC was produced, but lack of water supply limited physical leaching and so it was subsequently consumed by microbes present in the samples (Moore and Dalva, 2001). The findings presented here, that CO₂:DOC partitioning from *Sphagnum* and peat could be affected by climate change, but that the partitioning from other substrates was not affected, will be of interest to those modelling the response of peatland C balance as it indicates that DOC flux cannot be estimated from ecosystem respiration alone. It is also useful information for water treatment engineers who are trying to plan future climate-proof investments in treatment works, as these results imply that DOC could increase or decrease in the future, depending on the proportion of different plant species in the catchments, and the pattern of rainfall.

432

4.3. *Effects and interactions between temperature and rainfall*

Results indicate that simulated temperature changes have a greater effect on the loss of DOC (generally decreasing the flux) and on mass loss (generally increasing the flux) than on the production of CO₂. The effect varied between substrates, for example, drier conditions increased the DOC flux from *Calluna*, but decreased it from *Sphagnum*. When looking at the main drivers of increased decomposition in peatlands, other laboratory based studies have also found mixed results. Higher temperatures led to increased mass loss (between 5, 10 and 15 °C) in a one year incubation of Scots pine needles below a living moss layer (Domisch et al., 2006), supporting the results found here. Neff and Hooper (2002) also found that temperature controlled the CO₂ flux from their incubated soil samples, but the response depended largely

443 on vegetation community. However, in their study on DOC production across a range of spatial
444 scales, Preston et al. (2011) found that temperature had no effect on DOC production in their
445 peat microcosm experiment, regardless of moisture treatment. This contrasts with Clark et al.
446 (2006) who found increased DOC release at 20 °C relative to 10 °C in another laboratory study
447 incubating peat mesocosms.

448 Whilst measured mass loss did increase positively with temperature, measured CO₂
449 fluxes did not. Investigations at lower temperatures than those used here have suggested
450 *Sphagnum* becomes more productive, growing faster, with increases in temperature from a
451 low base (Breeuwer et al., 2008), but that the opposite can occur for warmer temperatures
452 (17.5 and 21.7 °C) (Breeuwer et al., 2009). Results herein indicate that decomposition,
453 measured through mass loss, increased with temperature, suggesting that the strength of the
454 peatland C sink could be reduced during summer months.

455 Rainfall treatment was just above the $P = 0.05$ critical threshold for cumulative CO₂ (P
456 = 0.075) and DOC flux ($P = 0.070$), with lower values seen under the UKCP09 (lower rainfall)
457 conditions. For DOC in peat, this agrees with the findings of Tang et al. (2013) who found that
458 DOC export was greater in the wetter treatments, and declined in the drought treatments.
459 Observed responses of DOC export to drought or experimentally lowered water tables have
460 been varied, with suggestions that DOC export could increase due to drought (Worrall et al.,
461 2006). However, there are mechanisms which could explain the decrease in DOC observed
462 here. Decreased DOC during droughts could be due to a suppression of DOC transport by
463 drought-induced decrease in DOC solubility (Clark et al., 2012, 2006), or an increase in
464 hydrophobicity (Worrall et al., 2008) rather than a change in the production of DOC through
465 organic matter decomposition. A difference here is that this study simulated rainfall whereas

those examining the mechanisms have tended to focus on water table effects, either through manipulations in the laboratory or event-based analysis in the field.

4.4. *Limitations of the study*

The data presented here are a useful indication of the relative importance of different substrates as sources of DOC and CO₂ fluxes, decomposition and the potential for changes in partitioning between CO₂ and DOC under future climate conditions. However, these are laboratory based simulation experiments and are subject to constraints that must be taken into account when interpreting the results. Samples may have been exposed to more extreme drying conditions than under field conditions as moisture levels were not regulated by living vegetation and/or underlying water table, and the lack of a peat substrate beneath the samples from the vegetation groups could also have increased the loss of water and so negated any rainfall treatment effect. Ritson et al. (2017) found that any exposure to oxygen (during laboratory experiments) increased DOC from peat and changed the quality of the C. The differences with the results presented here suggest a high sensitivity to the amount and frequency of the rainfall treatments. The findings of these laboratory experiments now need to be tested under field conditions.

4.5. *Wider implications for our understanding of peatland carbon balances*

While predictions about future rainfall patterns are variable, with both increases and decreases projected for northern latitudes (IPCC, 2014), models are in strong agreement that temperatures at mid-to-high latitudes will increase due to climate change (Murphy et al., 2009), and that drought events are likely to become more frequent (Meehl and Tebaldi, 2004).

489 While there are differences in both magnitude and direction of change between the
490 substrates, the data here suggest that warmer summer temperatures and decreased rainfall
491 could increase overall decomposition, reduce the flux of DOC, and subsequently alter the
492 partitioning of carbon released during decomposition from DOC to CO₂.

493 Other studies have suggested that the effect of climate change will be more through
494 indirect effects on the composition of vegetation, with graminoids and dwarf shrubs being
495 favoured over *Sphagnum* mosses in warmer, drier conditions (Kapfer et al., 2011; Weltzin et
496 al., 2003). Given the greater lability of the litter produced by these plant functional types, seen
497 here in the greater decomposition of the *Molinia* samples, this could be a concern for climate
498 change mitigation. However, in SW England, where these samples were collected, graminoids
499 (*Molinia*) are already highly prevalent (Chambers et al., 1999) and so the effect may not be as
500 severe, as climate-induced vegetation change may already have occurred. The shallow peat in
501 this area suggests the area could have been sequestering C at a lower rate than other blanket
502 peatlands, or even losing C, though further work looking at the entire C balance is needed to
503 verify this.

504 In the field, it is likely that climate change may increase productivity, and therefore litter
505 production, alongside any increase in decomposition rates. Studies have already provided
506 evidence that warmer temperatures can increase rates of carbon sequestration at some
507 northern latitude peatlands, due in part to longer and warmer growing seasons enhancing
508 plant growth (Loisel and Yu, 2013). Therefore, when interpreting the differences in
509 decomposition rates of the different substrates in the context of *in situ* actual conditions, it is
510 also important to consider their respective litter production rates and substrate supply (Ritson
511 et al., 2016). While *Molinia* showed the fastest decomposition in this experiment, it also tends
512 to produce more litter than *Calluna* (e.g. Aerts, 1989), so is likely to be a more significant

513 source of C at the catchment scale. Also, while both substrates generally produce less “litter”
514 than *Sphagnum* (Bragazza et al., 2012), this may change in a warmer, drier climate (Breeuwer
515 et al., 2009). Further work is needed to take into account the effect of climate change on
516 productivity and litter production, as well as on decomposition, to assess the likely impacts of
517 climate change on the future carbon balance of peatlands.

518

519 5. Conclusions

520 This study has used short-term laboratory incubations of peat and peat-forming
521 vegetation from a climatically marginal blanket peatland in SW England in order to assess the
522 relative importance of different temperature and rainfall scenarios for the main C fluxes (CO₂
523 and DOC) and their partitioning during short-term litter decay. The largest variation in C fluxes
524 and total mass loss was seen between substrates, with smaller fluxes (cumulative CO₂ and
525 DOC) and mass lost from *Sphagnum*/peat compared to higher fluxes and mass loss from dwarf
526 shrub/graminoids. The CO₂:DOC ratio was lowest for *Molinia* and highest for *Sphagnum*.
527 Climate factors were important too, but the effect depended largely on the substrate. For
528 example, while the future (drier) rainfall scenario increased the DOC flux from *Calluna*, it had
529 the opposite effect for *Sphagnum*. Therefore, indirect effects on species composition in
530 peatlands could ultimately turn out to be more important than direct effects of climate change
531 from increased temperatures and decreased rainfall.

532 Upscaling of this work from laboratory to field is needed in order to confirm these
533 laboratory-based findings and improve understanding of the likely impacts of climate change
534 on C fluxes and decomposition in peatlands. Further work across natural climatic gradients is
535 one way of addressing this research need, deploying a space for time substitution to
536 understand future climate change.

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546

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548 **References**

- 549 1. Aerts, R., 1989. Aboveground Biomass and Nutrient Dynamics of *Calluna vulgaris* and
550 *Molinia caerulea* in a Aboveground biomass and nutrient dynamics of *Calluna vulgaris*
551 and *Molinia caerulea* in a dry heathland. *OIKOS* 56, 31–38.
- 552 2. Billett, M., Charman, D., Clark, J., Evans, C., Evans, M., Ostle, N., Worrall, F., Burden,
553 A., Dinsmore, K., Jones, T., McNamara, N., Parry, L., Rowson, J., Rose, R., 2010.
554 Carbon balance of UK peatlands: current state of knowledge and future research
555 challenges. *Clim. Res.* 45, 13–29. doi:10.3354/cr00903
- 556 3. Billett, M.F., Palmer, S.M., Hope, D., Deacon, C., Storeton-West, R., Hargreaves, K.J.,
557 Flechard, C., Fowler, D., 2004. Linking land-atmosphere-stream carbon fluxes in a
558 lowland peatland system. *Global Biogeochem. Cycles* 18, 1–12.
559 doi:10.1029/2003GB002058
- 560 4. Blok, D., Elberling, B., Michelsen, A., 2015. Initial Stages of Tundra Shrub Litter

- 561 Decomposition May Be Accelerated by Deeper Winter Snow But Slowed Down by
562 Spring Warming. *Ecosystems* 19, 155–169. doi:10.1007/s10021-015-9924-3
- 563 5. Bonnett, S.A.F., Ostle, N., Freeman, C., 2006. Seasonal variations in decomposition
564 processes in a valley-bottom riparian peatland. *Sci. Total Environ.*, 370, 561–573.
- 565 6. Bragazza, L., Buttler, A., Habermacher, J., Brancaleoni, L., Gerdol, R., Fritze, H.,
566 Hanajík, P., Laiho, R., Johnson, D., 2012. High nitrogen deposition alters the
567 decomposition of bog plant litter and reduces carbon accumulation. *Glob. Chang.*
568 *Biol.* 18, 1163–1172. doi:10.1111/j.1365-2486.2011.02585.x
- 569 7. Bragazza, L., Buttler, A., Robroek, B.J.M., Albrecht, R., Zaccane, C., Jassey, V.E.J.,
570 Signarbieux, C., 2016. Persistent high temperature and low precipitation reduce peat
571 carbon accumulation. *Glob. Chang. Biol.* 22, 4114–4123. doi:10.1111/gcb.13319
- 572 8. Bragazza, L., Buttler, A., Siegenthaler, A., Mitchell, E.A.D., 2009. Plant Litter
573 Decomposition and Nutrient Release in Peatlands, in: Baird, A.J., Belyea, L.R., Comas,
574 X., Reeve, A.S., Slater, L. (Eds.), *Carbon Cycling in Northern Peatlands*. American
575 Geophysical Union, pp. 99–110.
- 576 9. Breeuwer, A., Heijmans, M.M.P.D., Gleichman, M., Robroek, B.J.M., Berendse, F.,
577 2009. Response of *Sphagnum* species mixtures to increased temperature and
578 nitrogen availability. *Plant Ecol.* 204, 97–111. doi:10.1007/s11258-009-9571-x
- 579 10. Breeuwer, A., Heijmans, M.M.P.D., Robroek, B.J.M., Berendse, F., 2008. The effect of
580 temperature on growth and competition between *Sphagnum* species. *Oecologia* 156,
581 155–67. doi:10.1007/s00442-008-0963-8
- 582 11. Broder, T., Blodau, C., Biester, H., Knorr, K.H., 2012. Peat decomposition records in
583 three pristine ombrotrophic bogs in southern Patagonia. *Biogeosciences* 9, 1479–
584 1491. doi:10.5194/bg-9-1479-2012

- 585 12. Bubier, J.L., Moore, T.R., Bledzki, L. a., 2007. Effects of nutrient addition on
586 vegetation and carbon cycling in an ombrotrophic bog. *Glob. Chang. Biol.* 13, 1168–
587 1186. doi:10.1111/j.1365-2486.2007.01346.x
- 588 13. Buttler, A., Robroek, B.J.M., Laggoun-Défarge, F., Jassey, V.E.J., Pochelon, C., Bernard,
589 G., Delarue, F., Gogo, S., Mariotte, P., Mitchell, E.A.D., Bragazza, L., 2015.
590 Experimental warming interacts with soil moisture to discriminate plant responses in
591 an ombrotrophic peatland. *J. Veg. Sci.* 26, 964–974. doi:10.1111/jvs.12296
- 592 14. Chambers, F.M., Mauquoy, D., Todd, P.A., 1999. Recent rise to dominance of *Molinia*
593 *caerulea* in environmentally sensitive areas : new perspectives from palaeoecological
594 data. *J. Appl. Ecol.* 36, 719–733.
- 595 15. Clark, J., Gallego-Sala, A., Allott, T., Chapman, S., Farewell, T., Freeman, C., House, J.,
596 Orr, H., Prentice, I., Smith, P., 2010. Assessing the vulnerability of blanket peat to
597 climate change using an ensemble of statistical bioclimatic envelope models. *Clim.*
598 *Res.* 45, 131–150. doi:10.3354/cr00929
- 599 16. Clark, J.M., Chapman, P.J., Heathwaite, A.L., Adamson, J.K., 2006. Suppression of
600 Dissolved Organic Carbon by Sulfate Induced Acidification during Simulated Droughts.
601 *Environ. Sci. Technol.* 40, 1776–1783. doi:10.1021/es051488c
- 602 17. Clark, J.M., Heinemeyer, A., Martin, P., Bottrell, S.H., 2012. Processes controlling DOC
603 in pore water during simulated drought cycles in six different UK peats.
604 *Biogeochemistry* 109, 253–270. doi:10.1007/s10533-011-9624-9
- 605 18. Dieleman, C.M., Branfireun, B.A., McLaughlin, J.W., Lindo, Z., 2015. Climate change
606 drives a shift in peatland ecosystem plant community: Implications for ecosystem
607 function and stability. *Glob. Chang. Biol.* 21, 388–395. doi:10.1111/gcb.12643
- 608 19. Dinsmore, K.J., Billett, M.F., Dyson, K.E., 2013. Temperature and precipitation drive

- temporal variability in aquatic carbon and GHG concentrations and fluxes in a peatland catchment. *Glob. Chang. Biol.* 19, 2133–48. doi:10.1111/gcb.12209
20. Dinsmore, K.J., Billett, M.F., Skiba, U.M., Rees, R.M., Drewer, J., Helfter, C., 2010. Role of the aquatic pathway in the carbon and greenhouse gas budgets of a peatland catchment. *Glob. Chang. Biol.* 16, 2750–2762. doi:10.1111/j.1365-2486.2009.02119.x
21. Domisch, T., Finér, L., Laine, J., Laiho, R., 2006. Decomposition and nitrogen dynamics of litter in peat soils from two climatic regions under different temperature regimes. *Eur. J. Soil Biol.* 42, 74–81. doi:10.1016/j.ejsobi.2005.09.017
22. Evans, C.D., Freeman, C., Cork, L.G., Thomas, D.N., Reynolds, B., Billett, M.F., Garnett, M.H., Norris, D., 2007. Evidence against recent climate-induced destabilisation of soil carbon from ^{14}C analysis of riverine dissolved organic matter. *Geophys. Res. Lett.* 34, L07407. doi:10.1029/2007GL029431
23. Fenner, N., Freeman, C., Ostle, N.J. and Reynolds, B., 2004. Peatland carbon efflux partitioning reveals that sphagnum photosynthate contributes to the DOC pool. *Plant and Soil.* 259, 345–354.
24. Fenner, N., Ostle, N.J., McNamara, N., Sparks, T., Harmens, H., Reynolds, B., Freeman, C., 2007. Elevated CO_2 Effects on Peatland Plant Community Carbon Dynamics and DOC Production. *Ecosystems* 10, 635–647. doi:10.1007/s10021-007-9051-x
25. Frolking, S., Roulet, N.T., Tuittila, E., Bubier, J.L., Quillet, A., Talbot, J., Richard, P.J.H., 2010. A new model of Holocene peatland net primary production, decomposition, water balance, and peat accumulation. *Earth Syst. Dyn.* 1, 1–21. doi:10.5194/esd-1-1-2010
26. Freeman, C., Lock, M.A., Reynolds, B., 1993. Fluxes of carbon dioxide, methane and nitrous oxide from a Welsh peatland following simulation of water table draw-down:

- Potential feedback to climatic change. *Biogeochemistry*. 19, 51–60.
27. Gatis, N., Luscombe, D.J., Grand-Clement, E., Hartley, I.P., Anderson, K., Smith, D., Brazier, R.E., 2015. The effect of drainage ditches on vegetation diversity and CO₂ fluxes in a *Molinia caerulea*-dominated peatland. *Ecohydrology* n/a-n/a. doi:10.1002/eco.1643
28. Gorham, E., 1991. Northern Peatlands: Role in the carbon cycle and probable responses to climatic warming. *Ecol. Appl.* 1, 182–195.
29. Grand-Clement, E., Anderson, K., Smith, D., Luscombe, D., Gatis, N., Ross, M., Brazier, R.E., 2013. Evaluating ecosystem goods and services after restoration of marginal upland peatlands in South-West England. *J. Appl. Ecol.* n/a-n/a. doi:10.1111/1365-2664.12039
30. Grand-Clement, E., Luscombe, D.J., Anderson, K., Gatis, N., Benaud, P., Brazier, R.E., 2014. Antecedent conditions control carbon loss and downstream water quality from shallow, damaged peatlands. *Sci. Total Environ.* 493, 961–73. doi:10.1016/j.scitotenv.2014.06.091
31. Harrison, A.F., Taylor, K., Scott, A., Poskitt, J., Benham, D., Grace, J., Chaplow, J., Rowland, P., 2008. Potential effects of climate change on DOC release from three different soil types on the Northern Pennines UK: examination using field manipulation experiments. *Glob. Chang. Biol.* 14, 687–702. doi:10.1111/j.1365-2486.2007.01504.x
32. Holden, J., Burt, T.P., 2002. Infiltration, runoff and sediment production in blanket peat catchments: implications of field rainfall simulation experiments. *Hydrol. Process.* 16, 2537–2557. doi:10.1002/hyp.1014
33. Hope, D., Billett, M.F., Cresser, M.S., 1994. A review of the export of carbon in river

- 657 water: Fluxes and processes. *Environ. Pollut.* 84, 301–324.
- 658 34. IPCC, 2014. IPCC, 2014: Climate Change 2014: Synthesis Report. Contribution of
659 Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental
660 Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)].
661 Geneva, Switzerland.
- 662 35. Johnson, L.C., Damman, A.W.H., 1991. Species Controlled Sphagnum Decay on a
663 South Swedish raised bog Sphagnum decay. *Oikos* 61, 234–242.
- 664 36. Kalbitz, K., Geyer, S., 2002. Different effects of peat degradation on dissolved organic
665 carbon and nitrogen. *Org. Geochem.* 33, 319–326. doi:10.1016/S0146-
666 6380(01)00163-2
- 667 37. Kapfer, J., Grytnes, J.-A., Gunnarsson, U., Birks, H.J.B., 2011. Fine-scale changes in
668 vegetation composition in a boreal mire over 50 years. *J. Ecol.* 99, 1179–1189.
669 doi:10.1111/j.1365-2745.2011.01847.x
- 670 38. Keselman, H.J., 1975. A Monte Carlo investigation of three estimates of treatment
671 magnitude: Epsilon squared, eta squared, and omega squared. *Can. Psychol. Rev.* 16,
672 44–48.
- 673 39. Keuskamp, J.A., Dingemans, B.J.J., Lehtinen, T., Sarneel, J.M., Hefting, M.M., 2013.
674 Tea Bag Index: a novel approach to collect uniform decomposition data across
675 ecosystems. *Methods Ecol. Evol.* 4, 1070–1075. doi:10.1111/2041-210X.12097
- 676 40. Koehler, A.-K., Sottocornola, M., Kiely, G., 2011. How strong is the current carbon
677 sequestration of an Atlantic blanket bog? *Glob. Chang. Biol.* 17, 309–319.
678 doi:10.1111/j.1365-2486.2010.02180.x
- 679 41. Laine, A., Sottocornola, M., Kiely, G., Byrne, K.A., Wilson, D., Tuittila, E.-S., 2006.
680 Estimating net ecosystem exchange in a patterned ecosystem: Example from blanket

- 681 bog. Agric. For. Meteorol. 138, 231–243.
- 682 42. Limpens, J., Berendse, F., 2003. How litter quality affects mass loss and N loss from
683 decomposing Sphagnum. Oikos 103, 537–547. doi:10.1034/j.1600-
684 0706.2003.12707.x
- 685 43. Limpens, J., Berendse, F., Blodau, C., Canadell, J.G., Freeman, C., Holden, J., Roulet,
686 N., Rydin, H., Schaepman-Strub, G., 2008. Peatlands and the carbon cycle: from local
687 processes to global implications - a synthesis. Biogeosciences Discuss. 5, 1379–1419.
- 688 44. Loisel, J., Yu, Z., 2013. Recent acceleration of carbon accumulation in a boreal
689 peatland, south central Alaska. J. Geophys. Res. Biogeosciences 118, 41–53.
690 doi:10.1029/2012JG001978
- 691 45. Luscombe, D.J., Anderson, K., Gatis, N., Grand-Clement, E., Brazier, R.E., 2015. Using
692 airborne thermal imaging data to measure near-surface hydrology in upland
693 ecosystems. Hydrol. Process. 29, 1656–1668. doi:10.1002/hyp.10285
- 694 46. Meehl, G.A., Tebaldi, C., 2004. More intense, more frequent, and longer lasting heat
695 waves in the 21st century. Science 305, 994–7. doi:10.1126/science.1098704
- 696 47. Moore, T.R., Bubier, J.L., Bledzki, L., 2007. Litter Decomposition in Temperate
697 Peatland Ecosystems: The Effect of Substrate and Site. Ecosystems 10, 949–963.
698 doi:10.1007/s10021-007-9064-5
- 699 48. Moore, T.R., Dalva, M., 2001. Some controls on the release of dissolved organic
700 carbon by plant tissues and soils. Soil Sci. 166, 38–47.
- 701 49. Moore, T.R., Paré, D., Boutin, R., 2008. Production of Dissolved Organic Carbon in
702 Canadian Forest Soils. Ecosystems 11, 740–751. doi:10.1007/s10021-008-9156-x
- 703 50. Murphy, J.M., Sexton, D.M.H., Jenkins, G.J., Booth, B.B.B., Brown, C.C., Clark, R.T.,
704 Collins, M., Harris, G.R., Kendon, E.J., Betts, R.A., Brown, S.J., Humphrey, K.A.,

705 McCarthy, M.P., McDonald, R.E., Stephens, A., Wallace, C., Warren, R., Wilby, R.,
706 Wood, R.A., 2009. UK Climate Projections Science Report: Climate Change
707 Projections.

708 51. Neff, J.C., Hooper, D.U., 2002. Vegetation and climate controls on potential CO₂, DOC
709 and DON production in northern latitude soils. *Glob. Chang. Biol.* 8, 872–884.
710 doi:10.1046/j.1365-2486.2002.00517.x

711 52. Palmer, S.M., Hope, D., Billett, M.F., Dawson, J.J.C., Bryant, C.L., 2001. Sources of
712 organic and inorganic carbon in a headwater stream: Evidence from carbon isotope
713 studies. *Biogeochemistry* 52, 321–338. doi:10.1023/A:1006447706565

714 53. Parry, L.E., Charman, D.J., 2013. Modelling soil organic carbon distribution in blanket
715 peatlands at a landscape scale. *Geoderma* 211–212, 75–84.
716 doi:10.1016/J.GEODERMA.2013.07.006

717 54. Pastor, J., Solin, J., Bridgham, S.D., Updegraff, K., Harth, C., Weishampel, P., Dewey,
718 B., 2003. Global warming and the export of dissolved organic carbon from boreal
719 peatlands. *Oikos* 100, 380–386. doi:10.1034/j.1600-0706.2003.11774.x

720 55. Peuravuori, J., Ingman, P. and Pihlaja, K., 2003. Critical comments on accuracy of
721 quantitative determination of natural humic matter by solid state (13)C NMR
722 spectroscopy. *Talanta* 59(1), 177–89.

723 56. Preston, M.D., Eimers, M.C., Watmough, S.A., 2011. Effect of moisture and
724 temperature variation on DOC release from a peatland: conflicting results from
725 laboratory, field and historical data analysis. *Sci. Total Environ.* 409, 1235–42.
726 doi:10.1016/j.scitotenv.2010.12.027

727 57. R Core Team, 2015. R: A language and environment for statistical computing.

728 58. Ritson, J.P., Bell, M., Graham, N.J.D., Templeton, M.R., Brazier, R.E., Verhoef, A.,

- Freeman, C., Clark, J.M., 2014. Simulated climate change impact on summer dissolved organic carbon release from peat and surface vegetation: Implications for drinking water treatment. *Water Res.* 67, 66–76. doi:10.1016/j.watres.2014.09.015
59. Ritson, J.P., Bell, M., Brazier, R.E., Grand-Clement, E., Graham, N.J.D., Freeman, C., Smith, D., Templeton, M.R., Clark, J.M., 2016. Managing peatland vegetation for drinking water treatment. *Scientific Reports*. 6, 36751.
60. Ritson, J.P., Brazier, R.E., Graham, N.J.D., Freeman, C., Templeton, M.R., Clark, J.M., 2017. The effect of drought on dissolved organic carbon (DOC) release from peatland soil and vegetation sources. *Biogeosciences* 14, 2891–2902. doi:10.5194/bg-14-2891-2017
61. Rodwell, J., 1991. *British plant communities, Vol 2: mires and heaths*. Cambridge University Press, Cambridge.
62. Simpson, A.J., McNally, D.J., Simpson, M.J., 2011. NMR spectroscopy in environmental research: From molecular interactions to global processes. *Prog. Nucl. Magn. Reson. Spectrosc.* 58, 97–175. doi:10.1016/j.pnmrs.2010.09.001
63. Smith, J., Gottschalk, P., Bellarby, J., Chapman, S., Lilly, A., Towers, W., Bell, J., Coleman, K., Nayak, D., Richards, M., Hillier, J., Flynn, H., Wattenbach, M., Aitkenhead, M., Yeluripati, J., Farmer, J., Milne, R., Thomson, A., Evans, C., Whitmore, A., Falloon, P., Smith, P., 2010. Estimating changes in Scottish soil carbon stocks using ECOSSE. I. Model description and uncertainties. *Clim. Res.* 45, 179–192. doi:10.3354/cr00899
64. Straková, P., Penttilä, T., Laine, J., Laiho, R., 2012. Disentangling direct and indirect effects of water table drawdown on above- and belowground plant litter decomposition: consequences for accumulation of organic matter in boreal

753 peatlands. *Glob. Chang. Biol.* 18, 322–335. doi:10.1111/j.1365-2486.2011.02503.x

754 65. Tang, R., Clark, J.M., Bond, T., Graham, N., Hughes, D., Freeman, C., 2013.

755 Assessment of potential climate change impacts on peatland dissolved organic

756 carbon release and drinking water treatment from laboratory experiments. *Environ.*

757 *Pollut.* 173, 270–277.

758 66. Toberman, H., Evans, C.D., Freeman, C., Fenner, N., White, M., Emmett, B.A., Artz,

759 R.R.E., 2008. Summer drought effects upon soil and litter extracellular phenol oxidase

760 activity and soluble carbon release in an upland *Calluna* heathland. *Soil Biology &*

761 *Biochemistry* 40, 1519–1532.

762 67. Turetsky, M.R., 2003. The Role of Bryophytes in Carbon and Nitrogen Cycling.

763 *Bryologist* 106, 395–409.

764 68. van Breemen, N., 1995. How *Sphagnum* bogs down other plants. *Trends Ecol. Evol.*

765 10, 270–275.

766 69. Wallage, Z.E., Holden, J., 2011. Near-surface macropore flow and saturated hydraulic

767 conductivity in drained and restored blanket peatlands. *Soil Use Manag.* 27, 247–254.

768 doi:10.1111/j.1475-2743.2011.00336.x

769 70. Weltzin, J.F., Bridgham, S.D., Pastor, J., Chen, J., Harth, C., 2003. Potential effects of

770 warming and drying on peatland plant community composition. *Glob. Chang. Biol.* 9,

771 141–151. doi:10.1046/j.1365-2486.2003.00571.x

772 71. Wickham, H., 2009. *ggplot2: elegant graphics for data analysis*. Springer, New York.

773 72. Wickland, K.P., Neff, J.C., Aiken, G.R., 2007. Dissolved Organic Carbon in Alaskan

774 Boreal Forest: Sources, Chemical Characteristics, and Biodegradability. *Ecosystems*

775 10, 1323–1340. doi:10.1007/s10021-007-9101-4

776 73. Wieder, R.K., Lang, G.E., 1982. A Critique of the Analytical Methods Used in

777 Examining Decomposition Data Obtained From Litter Bags. *Ecology* 63, 1636–1642.

778 74. Wong, S., Hanna, J. V, King, S., Carroll, T.J., Eldridge, R.J., Dixon, D.R., Bolto, B.A.,

779 Hesse, S., Abbt-Braun, G., Frimmel, F.H., 2002. Fractionation of natural organic

780 matter in drinking water and characterization by ¹³C cross-polarization magic-angle

781 spinning NMR spectroscopy and size exclusion chromatography. *Environ. Sci.*

782 *Technol.* 36, 3497–503. doi:10.1021/es010975z

783 75. Worrall, F., Gibson, H.S., Burt, T.P., 2008. Production vs. solubility in controlling

784 runoff of DOC from peat soils – The use of an event analysis. *J. Hydrol.* 358, 84–95.

785 doi:10.1016/j.jhydrol.2008.05.037

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801 Tables and figure captions

802 **Table 1.** Organic matter properties for peatland vegetation and peat classes used in the
803 experiment. Values are means ($n = 5$) of subsample values, numbers in brackets are the
804 standard error of the mean. Aromaticity is a qualitative measure and was calculated from
805 cross polarization-magic angle spinning nuclear magnetic resonance (CP-MAS NMR).

Class	Dominant substrate	C (%)	N (%)	C:N ratio	Moisture content (g g dry wt ⁻¹)	Aromaticity (%)
<i>Calluna</i>	<i>Calluna vulgaris</i>	49.3 (0.02)	1.35 (0.01)	36.4 (0.22)	1.90 (0.01)	8
Mixed litter	Senesced <i>Molinia c</i>	45.9 (0.01)	1.29 (0.01)	35.6 (0.28)	5.38 (0.11)	5
<i>Molinia</i>	<i>Molinia caerulea</i>	45.5 (0.02)	2.34 (0.02)	19.5 (0.13)	3.56 (0.05)	5
Peat	Peat	29.1 (0.02)	1.64 (0.01)	17.7 (0.07)	6.06 (0.06)	11
<i>Sphagnum</i>	<i>Sphagnum</i> spp.	41.8 (0.03)	1.00 (0.01)	42.1 (0.28)	15.020.15)	7

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817 **Table 2.** ANOVA model results, where the model fitted is *response* = Temperature * Rainfall *
818 Substrate (Cumulative CO₂, DOC and CO₂:DOC ratio are square-root transformed data).
819 Significant differences ($P < 0.05$) are highlighted in bold and effect sizes (ω^2) are stated in
820 brackets. “Temp”, “Rain” and “Sub” are short for Temperature, Rainfall and Substrate,
821 respectively).

Variable	Factor	<i>F</i>	df	<i>P</i> value
Cumulative CO ₂ (mgCO ₂ -C gC ⁻¹)	Temperature	1.04	1	0.311
	Rainfall	3.25	1	0.075
	Substrate	631.08	4	< 0.001 (0.960)
	Temp:Rain	0.06	1	0.804
	Temp:Sub	1.27	4	0.288
	Rain:Sub	1.68	4	0.164
	Temp:Rain:Sub	1.27	4	0.290
Mass loss (mgCO ₂ -C gC ⁻¹)	Temperature	20.10	1	< 0.001 (0.005)
	Rainfall	0.01	1	0.927
	Substrate	946.23	4	< 0.001 (0.966)
	Temp:Rain	0.01	1	0.915
	Temp:Sub	3.80	4	0.007 (0.003)
	Rain:Sub	2.72	4	0.035 (0.002)
	Temp:Rain:Sub	0.56	4	0.690
DOC (mgCO ₂ -C gC ⁻¹)	Temperature	18.19	1	< 0.001 (0.015)
	Rainfall	3.38	1	0.070
	Substrate	247.14	4	< 0.001 (0.838)
	Temp:Rain	0.04	1	0.837
	Temp:Sub	5.12	4	0.001 (0.014)
	Rain:Sub	11.34	4	< 0.001 (0.035)
	Temp:Rain:Sub	4.43	4	0.003 (0.012)
CO ₂ -C:DOC-C	Temperature	15.57	1	< 0.001 (0.056)
	Rainfall	26.16	1	< 0.001 (0.097)
	Substrate	19.96	4	< 0.001 (0.292)
	Temp:Rain	0.01	1	0.921
	Temp:Sub	3.27	4	0.015 (0.035)
	Rain:Sub	8.71	4	< 0.001 (0.119)
	Temp:Rain:Sub	2.24	4	0.072

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824 **Figure 1.** Treatment effects on measures of carbon fluxes, and their ratio; (a) Cumulative CO₂
825 flux, (b) DOC flux, (c) Cumulative CO₂ to DOC ratio (Error bars indicate one standard error).
826 Different letters denote statistically significant differences in means between substrates and
827 asterisks denote significant differences between treatments for substrates based on Tukey
828 HSD test ($P < 0.05$).

829

830 **Figure 2.** Treatment effects on C loss estimated from mass and C quotient measurements
831 (Error bars indicate one standard error). Different letters denote statistically significant
832 differences in means between substrates and asterisks denote significant differences
833 between treatments for substrates based on Tukey HSD test ($P < 0.05$).

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